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FUNDAMENTALS OF UNDERGROUND WASTE DISPOSAL

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FUNDAMENTALS OF UNDERGROUND WASTE DISPOSAL*

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Waste materials from manufacturing processes are generally released, with or without treatment, into the nearest watercourse or into the atmosphere. Such operations should be carried on without nuisance and without impairment of the health and economy of the region. Because this is not always possible, and because industry recognizes its responsibilities to the region, some attention has been given to the use of the ground as another medium for disposal.

Discharge of liquid wastes into the ground may take place through shallow pits located above the ground water table or into shallow wells which penetrate usable potable or industrial ground waters. The probability of contamination from such practice is high, and many instances of chemical contamination are known. Another form of discharge is to inject the waste materials into connate waters located at relatively deep formations (from

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about 1,000 to 5,000 or more feet). The likelihood of contamination of potable water sources is considerably reduced by this technique, although some contamination has resulted from this method of operation. Unless care is exercised, the possibility also exists of contaminating valuable natural resources.

The practice of injection, probably started prior to 1925 with the return of oil field brines to the ground, was first cited by the U.S. Bureau of Mines (1). However, it was not until 1939 that salt water injection into sub-surface strata gained much prominence (2). In the East Texas field, this type of disposal was utilized in a large number of installations following an experimental project proving the feasibility of the method in 1936. Considerable literature has accumulated on the early use of this technique (3-7). Where conditions are satisfactory, salt water injection has been accepted by the industry as the most suitable method of disposal. The experience of the petroleum industry has led other groups, notably the chemical industry, to study and develop this method to meet its specific requirements. As a result, information is now available on the factors influencing ground disposal of chemical brines and wastes (8,9,10,34,38), chloromycetin (11), and phenolic wastes (12). Recently, the method has been suggested for the disposal of radioactive liquid wastes (13-20).

A review of industrial experience in sub-surface disposal of waste materials will help in the formulation of principles and guides leading to the promulgation of regulations for the discharge of radioactive materials into similar formations. In the discussion to follow, industrial experience will be reviewed and this experience will be applied to the disposal of radioactive wastes.

Industry Experience

Public Health Implications

The discharge of liquid waste materials into the ground has certain public health connotations. Of major concern is the contamination of potable and industrial ground water supplies. Aside from this aspect, there is also the economic one of destroying natural resources through the introduction of objectionable chemical constituents.

The possibility of ground water contamination is greatest when waste materials are discharged to surface pits overlying porous formations or into shallow wells penetrating fresh water aquifers. Considerable evidence is available on the contamination of ground waters from such practices (21-25). Billings (21) reports of the contamination of a city well at Douglas, Michigan from the discharge of chromates from a plating plant; the well was 1,000 ft distant from the point of waste discharge. Even with an acceleration in flow due to pumping of the well, it took three years for the waste material to reach the well. Similar chromate contamination has been reported in Nassau County, New York (22). At Lansing, Michigan, three years of pumping were required to exhaust picric acid contamination which had entered the ground waters from a disposal sump located 500 ft from the wells (21). Phenolic wastes, salts, gasoline, cheese factory wastes, and septic dry wells (21), as well as trichlorethylene (23), and oil field brines (24) also have been responsible for ground water contamination. Much of the reported literature on the movement of bacterial and chemical pollutants through the ground has been reviewed by Butler, Orlob, and McGauhey (25), who

conclude that chemical pollutants travel further than do bacterial pollutants.

Since in the use of ground disposal the possibility of ground water pollution does exist, probably the first requirement for an injection well is that it be cemented and thoroughly sealed below the potable water bearing formations. Another requirement is that no other wells pierce the injection formation within the volume that will be contaminated. Failure to observe this resulted in the abandonment initially of the well at the Dow Chemical plant in Midland, Michigan (8,9). The application of pressure to increase the rate of disposal increases the possibility of returning materials through abandoned or other wells in the vicinity.

Following a study of ground water pollution, a task group of the American Water Works Association (26) reported rules and regulations prepared with the assistance of an AWWA Michigan Section Committee for the Control of Ground Water Pollution. These regulations are based upon the following premises:

1. As contrasted with surface water pollution, ground water pollution can take place with little or no apparent evidence.
2. The time interval between initial waste disposal and the appearance of polluted wells may be so great as to permit irreparable damage to underground supplies.
3. The results of ground water pollution may be very longlasting - sometimes even to the extent of affecting future generations.
4. The direction and extent of ground water pollution are not easily identified, and are susceptible to the influences of artificial or natural changes in local hydrology.

Geological and Hydrological Factors

Since shallow disposal, either through surface pits or wells, con-

stitutes a definite hazard from the standpoint of pollution, the remainder of the discussion will be concerned with the discharge of materials into deep formations where the hazard is not as great. In selecting suitable sites for subterranean disposal, the first consideration is to find a formation which is sufficiently permeable to satisfy the physical requirements of injection. The types of formations that have been found suitable for the disposal of petroleum and chemical brines include sedimentary rock formations such as sandstones and limestones, and unconsolidated sands and gravel (2,29,30,31). For high injection rates, cavernous limestone formations appear most suitable, however, if channels are present, the waste materials may be conducted great distances and may lead to contamination of potable, ground, or surface waters.

Up to the present, the waste materials have been returned to brine deposits through injection wells located in Texas (10,30,34,38), Michigan (8,9,10,11,12), Oklahoma (10,30), and California (25). Pecsok (12), generalizing from a U.S. Geological Survey publication (32), states that disposal areas are most likely to be found in the central states bounded by the Appalachian Mountains on the east and the great plains region on the west. Most of this area is underlain by fairly horizontal sedimentary deposits and the deeper ground waters apparently have little movement. This section of the country, particularly in Michigan and Texas, is being used for deep disposal of various chemical wastes. Portions of this area are underlain by oil-bearing sands and brine deposits containing valuable chemicals and contamination of such resources must be avoided. However, where these sands have been exhausted of economic materials, they may serve for the disposal of wastes.

Chemical Considerations

The experience gained in the return of oil field brines and other chemical wastes to subsurface formations has demonstrated that the returned materials must be free of suspended solids, must be stable, sterile, and unreactive with natural brine and underground soil formations. The treatment practices used to produce suitable return wastes have been reported (2,7,8,9,10,11,31,34), and will be described. Jessen (2) has presented the most complete discussion of open and closed-type treatment and lists the various reactions that take place during the treatment steps. Similar discussions have been presented by Morris (7) and McRae (31). In the open-type system, treatment generally consists of alum coagulation followed by filtration and chlorination for the control of bacterial growths. Where the closed-type system is employed, hexametaphosphate treatment is frequently used to prevent precipitation of calcium and magnesium. Because of its oxidizing nature, chlorine cannot be used in the closed-type system as a bactericide.

At the Dow Chemical plant (8,9), a filter was used to pretreat the return brine to reduce the turbidity to less than 10 mg/l. A residue, consisting of a water solution of salts of organic acids from a fractionator separating formaldehyde, acetaldehyde, and methanol, was mixed with oil field brines and filtered to remove solids before return to a porous sand containing salt water 1900 ft below the surface (10). At a chemical plant in Oklahoma (10), a waste containing carbon particles and metallic hydroxides in suspension was passed through a sand filter before injection

to a limestone formation containing brine. The wastes from the Magnolia Petroleum Company plant, Fremont, Texas (10) are given chemical treatment and filtered before injection into the ground.

Adinoff (11) describes the treated chlorospecetin wastes available for injection as being strongly buffered with a pH of 3.7, as having a 20-day BOD of 45,000 mg/l and a dissolved solids content of 60,000 mg/l, as being very low in suspended solids since the waste consists essentially of filtrates and distillates, as having an intense reddish brown color equivalent to 12,000 by the color standard in "Standard Methods for the Examination of Water and Sewage," and as being toxic to Great Lake shiners at concentrations of 2 to 5 per cent. A tar-like residue precipitates out of the wastes slowly upon prolonged storing and aging. Because of this precipitation, large wooden tanks were installed to age and settle the wastes. These settled wastes were passed through a sand filter, thus reducing the possibility of plugging the porous limestone receiving formation.

Although preliminary tests with a synthetic brine showed compatibility with the phenolic waste materials to be returned (12), it was found that when mixed with the brine from the disposal formation the wastes were incompatible. The characteristics of the brine solution are given in Table 1. Considerable expenditure of effort resulted in the development of a suitable process (not described) to remove the resin-like ill-defined material believed to be responsible for the incompatibility of the waste and brine solution.

Table I. Characteristics of Brine Solution (Reichhold Chemical Co.)

Specific gravity - 20°C	1.1055
Ba	13.84%
Sodium Chloride	7.41%
Calcium chloride	3.63%
Magnesium chloride	1.59%
Potassium chloride	0.33%
Total chloride salts	12.92%
Total solids	16.1%
Pounds of sulfur per 1000 gal.	4.0
Sulfate and silica	Traces

Johnson (25) reports that a typical brine from the South California oil field in the vicinity of Torrence contains 11,000 mg/l of Na^+ , 17,000 mg/l of Cl^- , has a characteristic musty odor, and a total dissolved solids content of approximately 30,000 mg/l. He and Cecil (30) state that the chemical composition of the reservoir water is some multiple, perhaps fractional, of the chemical composition of sea water. Exceptions to this exist, and no formation should be used for disposal without an analysis of the reservoir water to study the reactions possible with the input water. Some reservoir waters contain considerable quantities of barium and strontium salts which will produce precipitates with sulfates or chromates. For example, Sudbury (35) reports connate waters having strontium concentrations up to 2,200 mg/l, barium of 900 mg/l, potassium of 1,400 mg/l, and lithium of 100 mg/l. The presence of 10 mg/l barium in the reservoir water was not considered high enough to interfere with the operations of the injection well at the Magnolia Petroleum Company (30). Calcium and magnesium chlorides in the reservoir water will produce precipitates of calcium carbonate and magnesium hydroxide with soluble carbonates and hydroxides. Such alkalies should be converted to a form which will not be precipitated. Treatment with carbon dioxide to convert them to bicarbonates will be better than conversion to mineral acid salts, and the minimum pH obtainable by carbonation will not be corrosive to a properly constructed disposal system.

One Oklahoma plant (30), using well injection for eight years, disposes of waste materials containing suspended matter consisting of carbon particles and some metallic hydroxides produced from corrosion of equipment. Before return to the salt water formation, the waste is sprayed on to a one-

foot gravity sand filter. The top sand is scraped daily and new sand added as needed.

The brine to be disposed of at Victoria, Texas, contains about 18 per cent sodium chloride with traces of metallic salts and organic compounds (34,35). It must be treated to prevent the formation of precipitates and it must be free of suspended solids. The metal content of the waste is reduced to about 20 mg/l.

As has been indicated earlier, stabilization of calcium salts to prevent precipitation of calcium sulfate and calcium carbonate is particularly important, and methods for predicting the tendency of oil field waters to deposit these two precipitates have been reported (35,36).

Economic Considerations

The cost of an injection well is dependent upon many factors including depth of well, degree of pretreatment, type of surface installation, permeability of the formation into which waste solution is returned, character of formation (sandstone or limestone), the quantity of brine handled, etc. The cost* of a closed-type system (2) for a capacity of 5,000 bbl

* The costs reflect prices at the time of the report. Some of these may have to be increased materially to correspond to present conditions.

(42 gal/bbl) per day is \$30,000, to which an operating cost not exceeding 1.5 cents per bbl should be added. This cost includes cleaning or backwashing the well. For larger installations, open-type systems handling 15,000 bbls of waste or more per day, the cost of treating plants will average

about \$30,000 while the cost of completing and equipping the disposal well is estimated at \$25,000. The operating cost again should not exceed 1.5 cents per bbl. Costs above and below these values are known and have been tabulated by Elliston and Davis (6).

Although well injection was abandoned at the Dow Chemical plant, Harlow (8) reports a cost of \$13 per 1,000 bbls of returning 43,000 bbls of waste per 24 hrs. On the Gulf Coast, costs for a 4-in inside diameter well 4,000 ft deep completely equipped ran to \$40, or \$10 per ft (10). This well handles 4,000 bbls of waste water daily. Capital investment amounts to about \$10 per bbl of daily capacity plus chemical treatment, which for either the closed or open system does not exceed 1.5 cents per bbl. Henkel (18) reports a cost of \$250,000 for the first well at the DuPont plant, Victoria, Texas, and indicates that their second well was constructed for considerably less.

Thermal Considerations

It is known that the temperature increases as one penetrates deeper into the crust of the earth. The effect of this temperature increase upon reaction rates between reservoir water and waste solution must be evaluated before return of the wastes. In some instances, increased temperatures will mean higher solution rates, whereas in others, as in the case of calcium carbonate, increased temperatures decrease solubilities, thus causing greater precipitation at the higher temperatures. The effect of injection pressures on temperature must also be taken into account, since at constant volume an increase in temperature is associated with an increase in pressure.

Ground Disposal of Radioactive Wastes

Public Health Implications

The premises outlined previously concerning the public health aspects of ground disposal of wastes apply at least as strongly to the disposal of radioactive waste materials into the ground. In reality, the problem poses additional difficulties in that the maximum permissible concentrations for many of the radionuclides of interest (27,28) when injected over a considerable period of time (70 years) are extremely low — many orders of magnitude lower than the permissible toxic levels of related chemical elements. Radioactive materials give off ionizing radiation which is damaging to tissue, and the length of time during which they may be damaging varies with the individual radionuclides. The half-life or decay time — time required for one-half of the radioactive atoms to disintegrate — may vary from a fraction of a second to millions of years. It cannot be changed by any physical, chemical, or biological method; only time being effective for reducing the level of activity. Another factor of importance is that the wastes generally contain a greater variety of chemical elements than are encountered in other industrial wastes, thus making treatment more difficult. Even though removed from the waste solution, these concentrates must still be disposed of in a safe manner to protect human and animal life from ionizing radiations. It has been suggested that the permissible levels defined for air and water (27,28) be reduced by a factor of 10 when applied to the general population (28).

Geological and Hydrological Considerations

The problem of disposal of radioactive wastes into the ground has been the subject of a recent Atomic Energy Commission sponsored meeting (33) and has been discussed in some detail by Theis (15), who states that the wastes may be placed in formations already saturated with natural liquid or that they may be placed in unsaturated ground high above the water table, as at Hanford (16). Geologically, the environment may consist of 1) sandy, relatively permeable formations, 2) permeable cavernous formations, 3) jointed or otherwise fractured rock, 4) relatively impermeable shales, or 5) deposits of salines — rock salt and gypsum. Any decision as to the use of the ground in any particular locality must be preceded by a consideration of the particular waste in its relation to the particular environment concerned.

A very important requirement of injection wells is that the hydraulics of its movement be understood in as great detail as possible, since movement depends upon the hydraulic properties of the formation and the contained liquid. Differences in specific gravity between the injected radioactive waste and the contained liquid will result in different movement of the two liquids. A similar situation will exist as a result of temperature differences between the two liquids.

The rate of flow of ground water must be well known before any radioactive wastes are discharged. If movement is slow, sufficient time may be available to permit decay of some of the radioisotopes; however, this is not effective for the decay of such long-lived materials as strontium-90 (20 year half-life) and cesium-137 (37 year half-life).

Injection into cavernous rocks, particularly in the presence of solution channels, would result in a more erratic and less predictable movement of the waste materials than in sandy formations. In limestone with this type of porosity, the permeability may be high and the porosity low, with the result that the wastes might spread more widely than in sandy formations.

Disposal into unsaturated unconsolidated material in arid regions, with a very deep water table, offers some possibilities. Such a situation exists at Hanford, Washington (16), where the water table is located about 100 meters below the surface. A potentially large reservoir exists for the adsorption and retention of radioisotopes, since the soil has an average exchange capacity of about 5 milliequivalents/100 gm of soil. Approximately 10^9 liters of active wastes containing several hundred thousand curies of fission products have been stored in such sites in the first decade of operations at Hanford (16).

Retention of radioactive liquid wastes in lined or unlined pits has been given some consideration. If located in suitable impervious formations, the need for a supplemental liner material is not as great as when these facilities are located in pervious formations. However, impervious formations are rather hard to find. Terrestrial pits have been in use at the Oak Ridge National Laboratory since 1951 for the disposal of intermediate level alkaline wastes containing radioactive cesium and ruthenium and small amounts of radioactive strontium. More than 35,000 curies of activity have been discharged to such open pits located in Conasauga shale (19) during the period August 1, 1951 through October 31, 1955. Experience has shown that

ruthenium, in the anionic or non-ionic form, moves through these formations as does the stable nitrate. To date there has been no evidence of any cations in any of the observation or sampling wells surrounding the waste pits. Artificial liners have been suggested for increasing the impermeability of surface pits, but no entirely satisfactory material that will be physically, chemically, thermally, or radiation stable has been found.

Of the various geological formations considered, salt deposits seem to offer the least hazard for the disposal of high level radioactive solutions. Cavities can be made by solution or mining. In general, no ground water movement occurs through salt; most, if not all, salt mines are dry even where they are far below the water table. They occur in many parts of the world and being plastic are more free than other rocks from earthquake hazard.

Chemical Characteristics

The chemical composition of radioactive wastes will be quite different from that of the connate brines normally encountered in the deeper formations. Depending upon the method used in processing the spent reactor fuel, the bulk of the waste material may be acid or alkaline and may contain large concentrations of aluminum and other materials. If the extraneous ions could be separated from the radionuclides, the volume of liquid for disposal would be quite small. Some of the physical and chemical characteristics of the wastes have been reported by Culler (37) and are summarized in Table 11.

Table II. Physical and Chemical Nature of the Wastes

Properties	Natural U - Pu		Enriched U-235
	Salted	HNO ₃ Salted	Diluent Salted
Boiling point	102° C.	112°C.	103°C.
Specific gravity	1.18	1.24	1.23
Concentration possible	3	?	2
Gal/gm U ²³⁵ Burned			
(Unneutralized)	2.5	0.5	5.0
Gal/gm U ²³⁵ Burned			
(Neutralized with NaOH)	10.0	1.5 to 2.0	20.0
Acidity, <u>N</u>	-0.3	8.0	-0.2
	(Basic)		
Total salts, <u>M</u> (Unneutralized)	1.2	0.2	2.0
Total salts, <u>M</u> (Neutralized)	6.0	8.2	Acid storage
Solids stability	Unstable	Stable	Unstable
	Basic		Basic

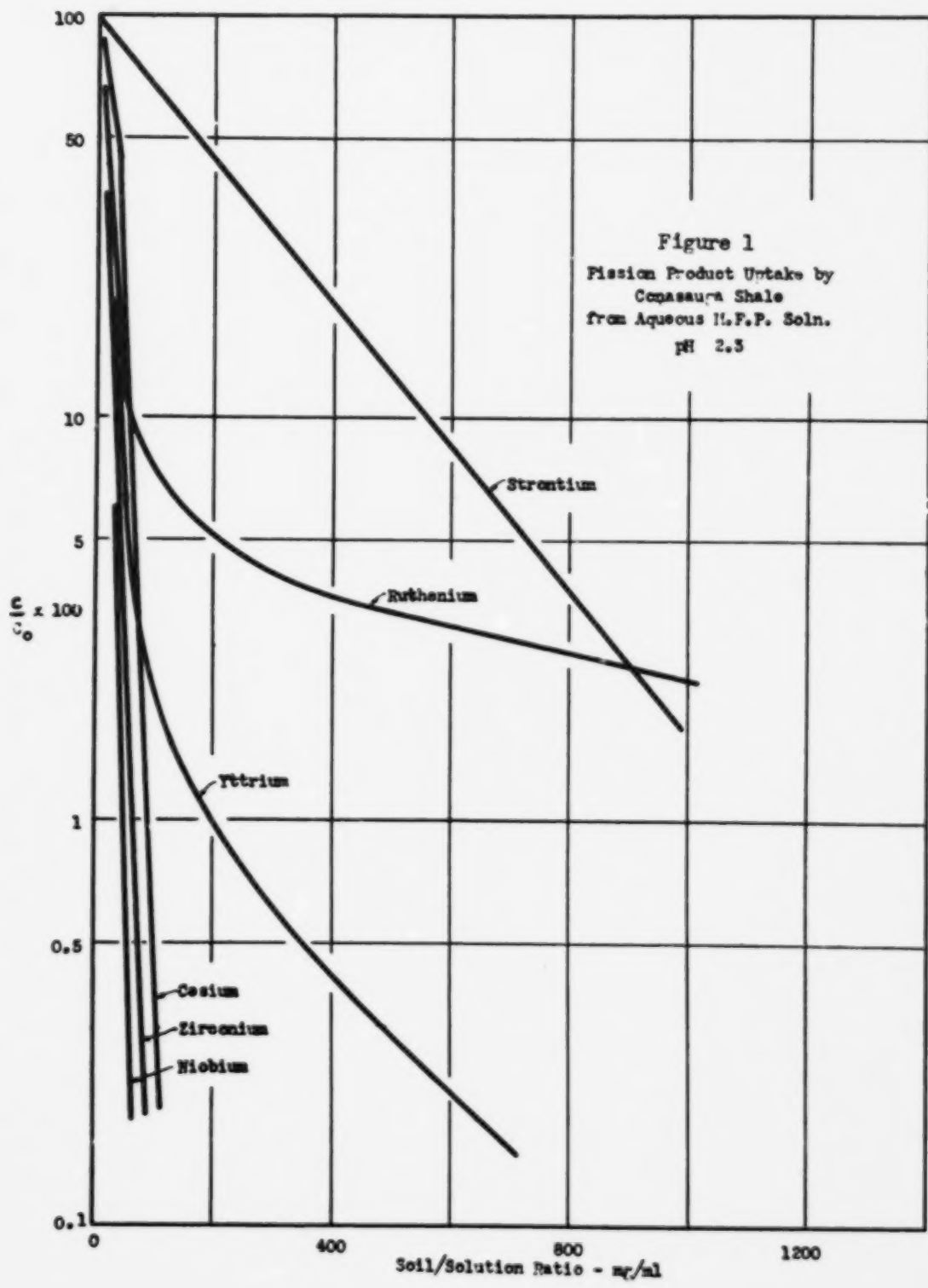
The solid salted process waste may be concentrated by a factor of 3 by evaporation, but this is the limit for the residue is then solidly set up. The volume of solids in some of these wastes after evaporation may be reduced by only a factor of 2 because of bulking.

Wastes may be stored as they come from the extraction tower, but some are neutralized, usually with sodium hydroxide, before storage. Due to the presence of amphoteric salts which require a large volume of neutralizing agent, the volume of the enriched U^{235} process wastes is increased approximately four-fold upon neutralization. As can be seen from the data presented, the natural uranium-plutonium salted process and the enriched U^{235} process wastes may be acid deficient, whereas the nitric acid salted system may contain 8 N nitric acid. Precipitates may form in a relatively short period of time after neutralization from either of the salted systems shown. Such precipitates as aluminum and iron hydroxides, which later convert to oxides and because they carry with them considerable quantities of radioactivity, make handling, storage, and disposal more difficult.

Wastes may be placed into the ground either as liquids or as solids. The resultant problems are different in each case. When placed into the ground as liquids, considerable information must be obtained on the reactions that take place between the introduced waste material and the reservoir material to avoid clogging the receiving formation, particularly if a porous formation is used for disposal. As these wastes move through the formation, there is an opportunity for exchange and/or adsorption between the ions contained in the waste and the reservoir medium and rock formation. The movement of these ions varies with the concentration of the ions present (radioactive and stable), with the pH of the resultant

solution, with the nature of the soil with which the waste materials are in contact, with the rate at which they are introduced into the ground, etc. Some of these effects have been defined by Kelley (39), particularly those factors affecting the cation exchange capacity of the soil. Considerable study is being given by the Oak Ridge National Laboratory and the Public Health Service to these factors as they relate to the movement of radioactive ions through local conasauga shale.

Typical results showing the movement of added radioactive tracer materials to a simulated waste solution and passing the mixture through Conasauga shale are shown in Figure 1. The waste did not contain aluminum nitrate and the pH was adjusted to various levels before contact with the soil column. It will be seen that there is considerable difference in the rate of movement of specific ions, and this movement is a function of the concentration of the various ions present, the valence state of the ion, and the pH of the medium. From the data shown, strontium, the most critical radionuclide requires the most soil for removal -- in other words it moves fastest through the soil mass. Ruthenium is removed quite rapidly, and then the rate is reduced. The rapid rate of removal may be associated with the removal of cationic ruthenium and the lower rate of removal with the conversion of anionic ruthenium (formed during the aging of the solution) back to the cationic ruthenium and its subsequent removal in this form. In general, it will be noted that with the exception of the cesium, the movement of ions is inversely proportional to the valence state of the ions, that is, the higher the valence state the more rapid the removal of the specific ion. Cesium is removed preferentially because it has a very high ionic radius, and once attached, it is removed with difficulty.

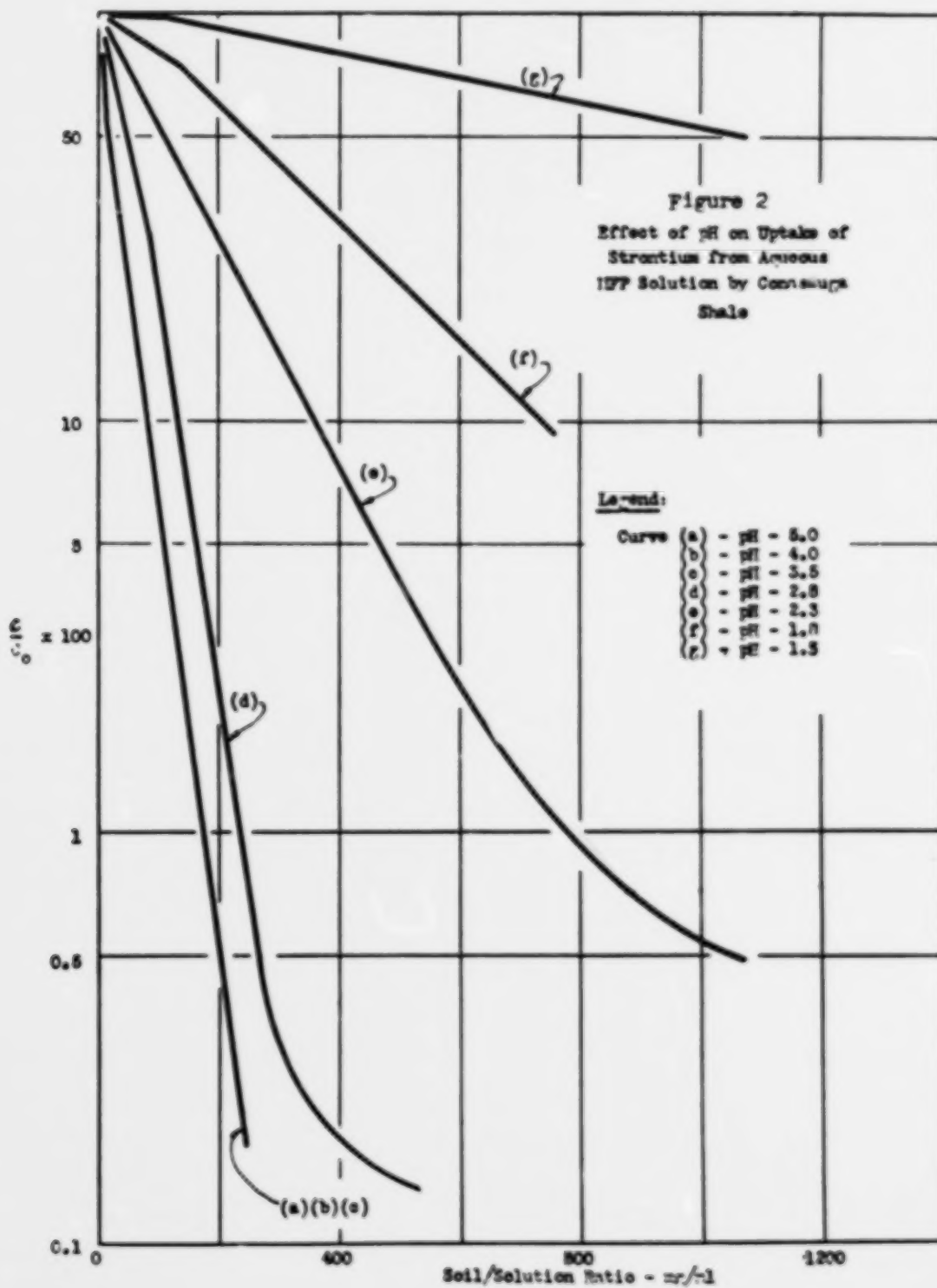


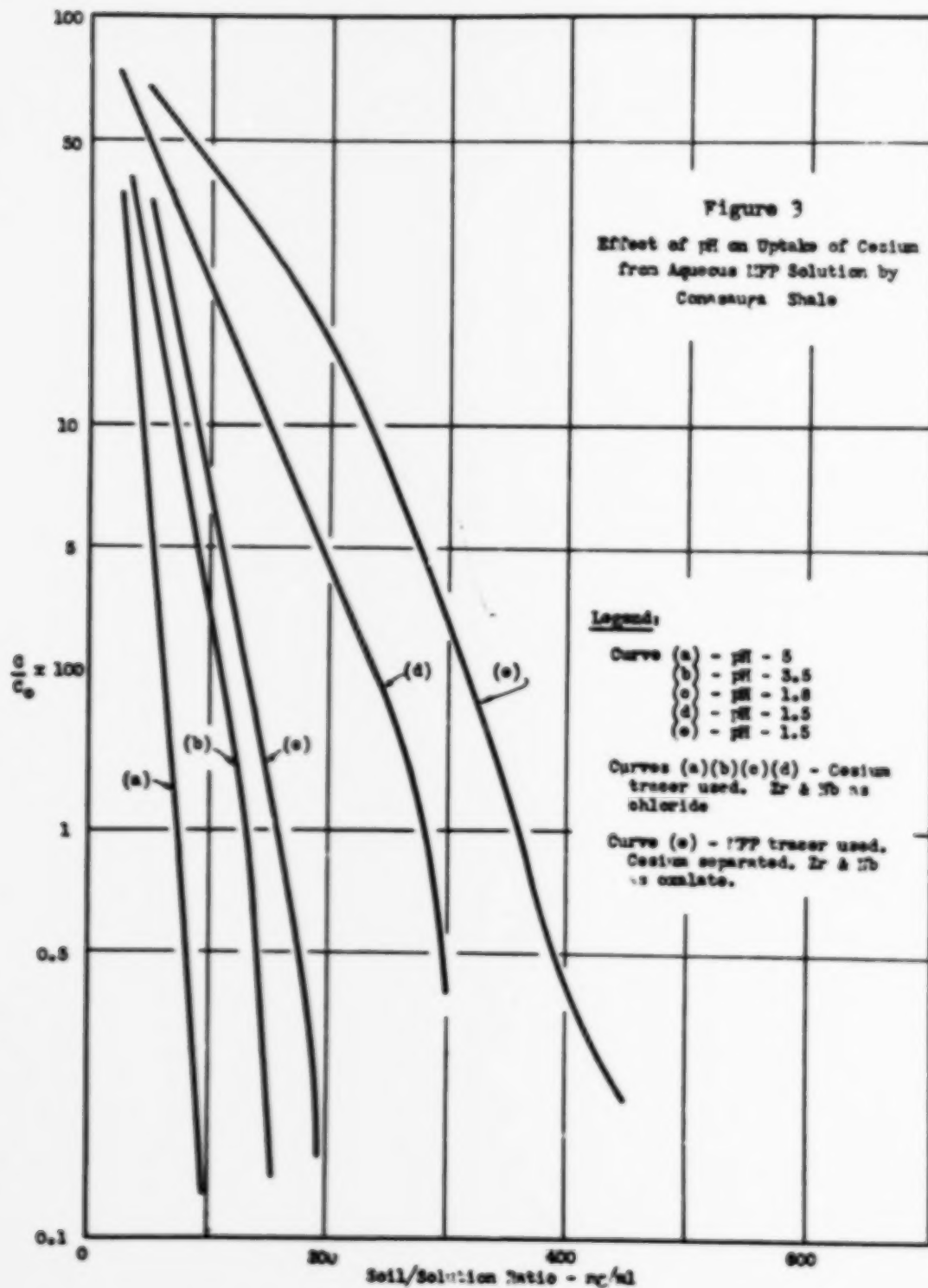
The effect of pH on the removal of strontium and cesium is shown in Figures 2 and 3. As the pH is reduced from 5 to 1.8, the amount of soil required per ml of solution to effect removal is increased. In the case of strontium (Figure 2), an increase in pH from 1.5 to 3.5 shows a very rapid change in the amount of soil required for removal; increasing the pH above 3.5 seems to have little effect upon the removal of strontium. Figure 3 shows similar findings in the case of cesium, but it will be seen that the amount of soil needed for removal is considerably less and the effect of lower pH values is less marked. Additional studies to define the effect of these and other factors or parameters mentioned earlier are needed.

The second case, that of the disposal of solid waste materials, where the radioactive ions have been incorporated in or fixed in a ceramic mass by firing, differs markedly from the disposal of liquid waste. Here, the basic problem is one of controlling the release of radioactive materials by leaching. If leaching can be restricted, the potential problem of ground water contamination by radionuclides is considerably reduced.

Economic Considerations

The cost of well injection, at least as reported by the petroleum industry was low, with operating costs of about 1.5 cents per bbl and fixed costs of about \$10 per ft for the injection well. These costs are extremely attractive when compared with the costs of providing tank storage for radioactive liquid wastes, which range from \$0.30 to \$2.00 per gal of tank capacity, and may be compared with the cost of providing surface pits for the disposal of intermediate radioactive wastes at 1.5 cents per gal of capacity. To either of these costs, the cost of monitoring, exploration, analysis, etc., must be added.





Thermal Effects

In the disposal of radioactive waste materials, consideration must be given to the heat liberated during decay of the radioactive ions. The amount of heat produced is a function of the energy of disintegration and is found to be equal to 5.9×10^{-3} E Watts per curie. The value of E for radioisotopes of intermediate half life is of the order of 1 Mev/dis, hence, a megacurie (10^6 curies) of radioactivity can be considered as emitting approximately 6 kilowatts or 336 BTU/min of energy.*

$$\begin{aligned} * \quad 1 \text{ curie} &= 1.33 \times 10^{11} \text{ dis/hr} \times 4.45 \times 10^{-17} \text{ watt-hr/Mev} \times E \text{ Mev/dis} = \\ &5.9 \times 10^{-3} E \text{ watts or } 5.9 \times 10^{-3} E \text{ watts} \times 5.692 \times 10^{-2} \frac{\text{BTU/min}}{\text{watt}} = 3.358 \times \\ &10^{-4} E \text{ BTU/min.} \end{aligned}$$

If sufficient radioactive materials are concentrated, boiling of the waste underground could take place and result in the escape of radioactive vapors to the surface. The computation of temperatures probably cannot be solved directly. With knowledge of the hydraulic constants of the formations, the exchange capacity of the formation materials for the radioactive components of the particular waste, and the heat generating characteristics of the waste, the temperatures can be computed by the assumption of heat sources of varying strength at proper spacing (15).

Perhaps the problem that will be most difficult of solution is the finding of suitable formations at or near projected nuclear power reactor stations provided with chemical processing facilities for recovering fissionable materials from spent fuel elements. If these are not located near-by, it will be necessary to transport highly radioactive liquids or solids in

which the radioactive materials have been immobilized to the disposal site. This will be very expensive because of the amount of shielding material that must surround the shipment to protect the public from external radiation hazards. At the reactor stations presently under construction by private enterprise and the AEC, the spent fuel elements will be shipped back to the National Laboratories for processing. The element of external hazard exists, but since the material is in a solid form, the danger of the release of these materials in case of an accident enroute is lessened.

Summary

The petroleum and chemical industries have employed ground disposal to their advantage for the disposal of their waste materials. The experience of these industries has been cited and extrapolated for possible use in the disposal of radioactive wastes. Unless a solution to this problem is found "... the accumulation of waste radioactivity from power reactors may very well place a limit on the benefits to be expected from nuclear power production." (1b).

2/2/53

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